# Landscape drivers of population density of a vulnerable apex predator (Wilkerr/dingo)

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# Abstract

Apex predators shape ecosystems globally, yet robust monitoring that assesses the effects of management actions and environmental variation on their populations is challenging. The dingo, Australia's largest terrestrial predator, is ecologically and culturally significant. In many parts of Australia, dingoes now exist in fragmented and isolated populations, and our understanding of how their population abundance and distribution is influenced by environmental and anthropogenic factors is limited. Here, we employed a flexible spatial mark–resight (SMR) modelling framework with strategically placed camera traps to estimate the density of an isolated and genetically distinct dingo population across a fire- and drought-prone, water-limited, semi-arid region of southern Australia. Our approach addressed detection challenges in remote landscapes and enabled the integration of key environmental covariates—including water availability, road proximity, and fire regime—to better understand spatial variation in dingo density. From over two million camera trap images, we identified 20 unique individuals at 38 locations using a combination of machine learning and manual validation. Dingo abundance and density was estimated as 77 (64–94) individuals and 0.014 (0.012–0.017) dingoes/km$^2$—lower than previously reported estimates—and varied significantly in response to aspects of the fire regime and distance from roads. Our findings highlight the value of integrating environmental covariates and resource-focused sampling strategies to improve detection, population estimates and inference of environmental preferences of large carnivores. Our approach can be adopted elsewhere to help inform management of landscapes and apex predator populations through robust population estimation of low-density carnivore populations in remote area contexts.

# Introduction

Apex predators perform critical roles in ecosystems (Ritchie and Johnson 2009, Ripple et al. 2014). Due to their high resource requirements, large carnivores typically occupy large territories and occur at low densities, and hence, management actions (e.g. lethal control), habitat modification (e.g. roads), and broader climate-and environmental-change can quickly drive changes in movements and/or result in declines in the distribution and population abundance of these species (Taylor and Goldingay 2010, Stier et al. 2016, Hradsky et al. 2017, Geary et al. 2020, Ordiz et al. 2021, Lee et al. 2024). As habitats and climate change, so too might key resources such as water or the likelihood and severity of key agents of disturbance such as fire. Shifts in space use due to changes in the availability of suitable habitat and resources can also intensify human–carnivore conflict (Woodroffe 2000, Ripple et al. 2014). It is therefore vital that we better understand such complexities for effective conservation and management of large carnivores.

Wildlife monitoring is crucial for protecting biodiversity and managing human-wildlife conflict, but because of their ecological traits, effective monitoring of large carnivores is often difficult and this, in turn, limits the efficacy of conservation and management efforts. Obtaining robust estimates of population density is inherently challenging (Murphy et al. 2019, 2024, Nolan et al. 2023), with spatial and/or temporal variability in environmental factors potentially further contributing to estimation biases (Schroeder et al. 2015, Supp et al. 2021, Hu et al. 2025). Understanding how environmental variables influence predator density has the potential to improve the accuracy of these estimates and, in turn, strengthen environmental policy and conservation outcomes.

As environmental and anthropogenic factors continue to shape predator distributions, reliable monitoring methods and adaptable modelling frameworks are becoming increasingly essential for understanding population dynamics and informing conservation strategies. Camera trapping has become an increasingly common non-invasive method for estimating species occurrence, abundance and to derive demographic patterns, distribution, and population densities for terrestrial wildlife (Miranda Paez et al. 2021, Bruce et al. 2025). This method can be particularly useful for large, low density, wide-ranging carnivores that are difficult to study with other methods (Forsyth et al. 2019, Gabriele-Rivet et al. 2020).

Camera-trapping surveys, combined with traditional capture–recapture or spatially explicit capture–recapture techniques, are widely used to estimate the density of individually identifiable carnivores. When only a subset of the population can be uniquely identified, spatial mark–resight models (SMR) offer a valuable alternative (Sollmann et al. 2013, Efford and Hunter 2018). SMR combine spatially explicit detection histories with survey area information to estimate population density. In the case of partially marked populations, SMR models integrate individual encounter histories of marked (identifiable) individuals with counts of unmarked (unidentifiable) individuals (Chandler and Royle 2013, Alonso et al. 2015, Whittington et al. 2018).

SMR models are also well-suited for investigating how habitat features, or environmental variables influence population distribution by modelling their effects on the spatial distribution of activity centres and detection probabilities (Efford 2004, Efford and Fewster 2013). By describing how individuals use and access different parts of the landscape, SMR models allow inference into behaviours such as mate seeking, home range dynamics, and resource tracking. This information enhances our understanding of how species respond to anthropogenic and environmental stochasticity, such as fire, water availability, and other spatial or temporal shifts in resource distribution (Payne et al. 2014, Geary et al. 2018), which is becoming increasingly importantly in a rapidly changing world (Johnson et al. 2023).

The dingo (*C. dingo/C. lupus dingo. Canis familiaris*, see (Cairns et al. 2025) is Australia's only large (>15kg) (non-human) native terrestrial apex predator (Letnic et al. 2012). Dingoes are widespread but patchily distributed across mainland Australia, and they are greatly reduced in number or have been extirpated from some regions (Fleming et al. 2001, Cairns et al. 2018). As a keystone predator they play a critical role in shaping ecosystems in Australia (Letnic et al. 2009b, 2012) and have been observed to exert top-down control on large herbivores such as kangaroos, wallabies and emus (Pople et al. 2000, Letnic and Koch 2010, Letnic et al. 2012, Dawson et al. 2023). They may also reduce the impact of introduced and/or feral mesopredators—red foxes and feral cats—through direct predation or fear-mediated changes to cat and fox spatial and temporal activity (Johnson and Vanderwal 2009, Letnic et al. 2009b, Brook et al. 2012, Colman et al. 2014, Geary et al. 2018), in turn indirectly protecting small-medium sized native prey (Johnson et al. 2007, Letnic et al. 2009a).

Despite the important ecological role of dingoes, their populations are subject to lethal control across much of mainland Australia, due to their perceived and actual impacts on the livestock industry (Allen and West 2013, Van Eeden et al. 2019, Campbell et al. 2022). These management practices do not always result in reduced dingo population sizes (Wallach et al. 2017, Stephens et al. 2023) and the removal of 'important' individuals can also lead to pack destabilisation causing a collapse of dingo social structures inadvertently increasing livestock predation (Allen 2014, 2015) and exacerbating conflicts over other shared resources, such as water (Wallach et al. 2009, O'Neill et al. 2017).

Comprehensive understanding of the dingo's ecological role in Australian ecosystems has been hindered by a severe lack of robust estimates of population abundance and density, and in relation to environmental variation. This challenge is especially pronounced for populations occurring at very low densities across vast, remote, and often inaccessible landscapes. Detecting such populations is difficult, even with camera traps—particularly when spatially random placement is used, which may be inefficient for sampling terrestrial predators whose movements and habitat use are typically non-random (Dickson et al. 2005, Newsome et al. 2013a, Murphy et al. 2024). In arid environments, water sources can serve as strategic focal points for camera deployment, particularly for water-limited predators such as dingoes (Allen 2012a, Newsome et al. 2013b). Targeting resource locations has been shown to improve detection rates and increase survey efficiency (Read et al. 2015, Edwards et al. 2016, Weijerman et al. 2024).

We undertook a landscape-scale analysis in a fire-prone, water-limited ecosystem in southern Australia to estimate the density of a wide-ranging, low-density carnivore: the dingo (*Canis familiaris*/*C. dingo*). Using a combination of resource-based and site-based camera traps, and a spatial mark–resight (SMR) modelling framework, we address the challenge of suspected low detection rates while accounting for both marked and unmarked individuals. To understand how key environmental variables influence dingo density, we modelled their effects on the spatial distribution of activity centres and detection probabilities, to inform management strategies. Maximizing detections while ensuring representative habitat sampling requires careful compromise in study design, balancing precision in abundance estimates with reliability in environmental inference. Due to the suspected extremely low density of dingoes in the region our design prioritised maximising detection probability while also attempting to capture environmental variation across the broader landscape.

Our specific objectives were to:

- Estimate dingo population size and density within the Big Desert Wilderness landscape.
- Understand how landscape features (water availability, fire regime and road density) affect dingo population size and density.
- Use these results to provide management recommendations for dingoes in the region.
- Provide broader insights for surveying and managing low-density carnivore populations in remote areas.

# Methods

## Study region

The Mallee region of Victoria's north is one of Australia's most fire-prone landscapes. This semi-arid, water limited region supports an isolated and genetically distinct population of dingoes (Wilkerr, the Wotjobaluk name) that was previously thought to be locally extinct (Weeks et al. 2025). Despite targeted monitoring efforts, this population has proven difficult to detect—likely due to its low density, large home ranges, and remote and largely inaccessible landscape.

Our *c.* 6,848 km² study area extends across a complex of conservation reserves in the Mallee region of north-western Victoria, encompassing Big Desert State Forest and Wilderness Area and Wyperfeld National Park, collectively referred to as “Big Desert Wilderness” (Fig. 1). The park complex occurs on the unceded, traditional lands of the Wotjobaluk and Ngarkat peoples. The region has a semiarid climate with mean annual rainfall ranging from approximately 267.9 mm in the north to around 370 mm in the south. Over the past 30 years (1989–2018), the region's average annual rainfall has decreased by about 20 mm (7%), from approximately 320 mm to 300 mm, compared to the previous 30 years (1959–1988) (Australian Bureau of Meteorology, 2024). Native vegetation within the reserves consists of “Mallee” vegetation consisting of Mallee eucalypt (*Eucalyptus leptophylla* and *Eucalyptus incrassata*) canopy with a heathy shrub understorey. “Heathland” vegetation generally lacks eucalypt trees and is instead dominated by a mixed layer of small (<2 m), heathy shrubs.

Fire has been a key ecological driver in this region, shaped over millennia by First Nations peoples, who applied fire to enhance productivity, aid in hunting, and for cultural purposes (Neale et al. 2019, McKemey et al. 2019, Fletcher et al. 2021). These practices maintained characteristic fire regimes, which were largely disrupted following British colonisation in the early to mid-1800s (Kimmerer and Lake 2001, Bowman et al. 2011). Post colonisation the region is characterised by large wildfires exceeding 100,000 hectares occurring approximately every 10–20 years (Pausas and Bradstock 2007, Avitabile et al. 2013), while smaller fire events occur more frequently (Gill and McCarthy 1998). Wildfire behaviour in mallee vegetation is typically characterized by uniform burn severity, resulting in large contiguous areas—often exceeding 10,000 hectares—experiencing a synchronized post-fire recovery stage. Because both wildfire and prescribed burns are generally stand-replacing and remove most above-ground vegetation, fuel accumulation occurs slowly, leading to prolonged fire intervals (Haslem et al. 2011) . As a result, the structure and composition of mallee vegetation are tightly coupled with fire regimes, which play a dominant role in shaping landscape heterogeneity and successional dynamics and therefore fauna habitat (Bradstock and Cohn 2002).

## Dingo data

We employed both systematic (site-based camera arrays) and strategic sampling (resource-based camera arrays) designs to ensure representative sampling of relevant environmental variables while also attempting to maximise dingo detection rates across the study area.

## Site - based arrays

An array of 289 cameras was deployed across sites primarily stratified by fire age class to represent the region's fire history: recently burnt ($n$ = 54), mid-successional ($n$ = 126), and late-successional ($n$ = 83). This distribution reflects the proportional extent of each fire age class across the landscape. In total 81 Swift Enduro wildlife monitoring cameras and 101 Reconyx (Reconyx H500) cameras over a period of three and a half years from November 2019 – March 2023 (Mason et al. 2025b, Pestell et al. 2025). Each camera was mounted on a wooden stake 50 cm above the ground. In front of each camera at approximately 3 m distance and within the field of view, scent lures were secured to an additional wooden stake baited with a combination of fish oil, anchovies and blood and bone to attract predators.

## Resource - based arrays

An additional 125 Swift Enduro wildlife monitoring cameras were placed at temporary water holes within Big Desert Wilderness area, from May 2023 to December 2024. Water is an important and limiting resource for dingoes, particularly in the arid zone (Newsome et al. 2013b, Wysong et al. 2020a). This approach is similar to carcass baiting (Sievert et al. 2023) and is suitable for sampling low density and wide-ranging animals (Nichols et al. 2019, Gabriele-Rivet et al. 2020, Harmsen et al. 2020, Carter et al. 2022).

## Image processing and final site selection

A total of 2,031,525 images were processed using a semi-automated approach. We used a combination of You Only Look Once (YOLO ) architecture for object detection and Microsoft MegaDetector for classification to species level (Redmon et al. 2015, Fennell et al. 2022). Once images were tagged as containing dingoes, manual inspection was used to identify individuals. Photographic catalogues were compiled for each uniquely identifiable dingo based on variation in pelage, distinctive markings, body size, scarring, and genetic deformities such as stumpy tails (Fig. 2).

Identification was only accepted if at least two distinctive features were visible (Fig. 2). As new individuals were encountered, they were added to the photographic catalogues. A double-blind protocol was applied, whereby only those dingoes independently identified by at least two researchers were included in the final dataset. Individual detection histories (presence/absence of an individual per location and 24-hour period beginning at 00:00) were constructed for each identifiable dingo. For those that could not be individually identified, we constructed a daily (24-hour) detection history for each uniquely named camera. We then selected the period with the highest number of known individuals and restricted the data from both site- and resource-based arrays to an approximate 12-month window (08/02/2024–15/01/2023) to minimise statistical issues associated with population turnover (immigration, emigration, recruitment), which can affect the reliability of population estimates (Allen et al., 2017; Balme et al., 2009; Royle et al., 2011). The final data set consisted of 7,923 survey days over 99 sites - consisting of three site-based and 96 resource based arrays (Table S1).

Figure 1. Location of site and resource-based camera arrays traps in Big Desert Wilderness Area. Panel a) indicates all sampled sites. Panel b) indicates sites where dingoes were detected within the 12-month survey period.

Figure 2. Three typical camera-trap images of dingoes (Wilkerr) in northwestern Victoria, Australia. Image a and image b indicate similar marked individuals which can be separated by length and shape of front leg ‘socks’. White chest pattern on the top individual is also extended while the button individual stops more abruptly. Image c contains three individuals with some patterning visible, however markings are not sufficient to reliably separate them.

## Landscape data

Dingo detections within the 12-month survey period were used to define the state space over which landscape variables were calculated for the SMR model. Given evidence that dingo movement and habitat use vary significantly across their range (Allen and West 2013, Newsome et al. 2017), we selected a 2.5 km resolution for all landscape layers to match the scale of likely space use and ensure ecological relevance. This resolution was chosen to capture variation in habitat features relevant to dingo ecology (Letnic and Dworjanyn 2011, Fleming et al. 2021) and is aligned with the spatial scales at which the species interacts with its environment (Brook et al. 2012, Fancourt et al. 2019).

## Distance to known water locations and roads

While rainfall in the Big Desert Wilderness region is generally seasonal there can be significant spatial and temporal variation across the landscape. During the 12-month survey period from December 31, 2022, to December 27, 2023, the Big Desert region in Victoria, Australia, experienced typical rainfall patterns with little to no recorded rainfall during April and May while the wettest period was August – September.

Water locations for the study area were derived from field-based data and remotely sensed imagery using a combination of deterministic layers and satellite-derived water observations (Geoscience Australia, 2024). Remotely sensed data was accessed via the Google Earth Engine (GEE) platform (Gorelick et al., 2017). Sentinel-2 Level 2A imagery (COPERNICUS/S2_SR_HARMONIZED) was used to assess surface reflectance over the 12-month survey period (31/12/2022 – 27/12/2023). Images were filtered to <10% cloud cover and further refined by masking clouds and cirrus (QA60 band). To capture seasonal variation in water availability and identify more permanent water bodies, imagery was grouped into four seasonal stacks: Summer (December–February), Autumn (March–May), Winter (June–August), and Spring (September–November). We applied a median composite to normalize illumination effects, computing median reflectance for each pixel in each stack (Mohammadimanesh et al. 2019). The final image sets included 16 spectral bands and 3 QA bands, from which we derived water and vegetation indices: Normalised Difference Water Index (NDWI) (McFeeters 1996), Modified Normalised Difference Water Index (MNDWI), and Modified Soil Adjusted Vegetation Index 2 (MSAVI-2) (Qi et al. 1994) Calculations for these indices are detailed in Table S2.

To supplement these data, we integrated Geoscience Australia's Water Observations, Multi-Year Frequency Statistics (WO-STATS, Landsat Collection 3, 30m, Frequency, 3.1.6), which provides water occurrence statistics from 1986 to the present. Finally, all data layers were imported into ArcGIS Pro v3.4, where known water locations were digitized using both field observations and high-resolution imagery. We derived a total of 37 known water locations across the state space.

Road data was derived from a publicly available Victorian government dataset from state agency records and included unsealed roads and tracks (Department of Transport and Planning, Victoria 2024).  These roads are typically unsealed, single-lane management tracks, with access primarily via the Murrayville-Nhill Track, which runs approximately north–south through the park. These tracks are not systematically maintained and can become overgrown with vegetation, making many parts of the park inaccessible.

We calculated distance to known water and distance to roads layers in ArcGIS Pro v3.4 using the Distance Accumulation tool. This produced continuous 30 m resolution rasters representing cumulative ‘cost’ surfaces from each feature, thereby characterising spatial variation of the surrounding landscape (Singleton et al. 2002). Distances were calculated from the center of each pixel, and subsequently resampled to 2.5 km resolution using bilinear interpolation, which also assumes cell-centred values, with the R package terra (Hijmans 2025). Distance to water (km) from digitized water sources provided a continuous measure of water accessibility within the state space. The mean distance to water was 12.87 km, with values ranging from 1.11 km to 43.86 km. This variation reflects the spatial distribution of water bodies across the landscape, where some areas had readily available water sources, while others were located farther from permanent water (Table 1). The Euclidean distance to roads within the study area varied widely, with a mean distance of 2.45 km and a range from 0.21 – 10.85 km, reflecting the remoteness and limited accessibility of the landscape (Table 1).

Table 1. Landscape variables describing disturbance and landscape fire used to calculate the response of dingo population size and density

| Variable | Ecological reason | Description | mean (min – max) | |
|---|---|---|---|---|
| | | | State space | Final camera locations |
| Distance to water (km) | Dingoes are water limited predators (Allen 2012a, Newsome et al. 2013b). The Big-Desert Wilderness is a semi- arid water limited region, and as such this resource is critical for the persistence of dingoes. | Distance Accumulation tool producing a continuous 'cost' raster | 12.87 km (1.11 – 43.86 km) | 4.09 km (1.11 – 32.01 km) |
| Distance to roads (km) | Infrastructure such as roads may be exploited by carnivores as travel corridors (Mercer et al. 2005, Dickson et al. 2005) while in other contexts they are avoided due to perceived risk (Kerley et al. 2002, Shepherd and Whittington 2006). As available habitat becomes increasingly fragmented, these shifts in space use can intensify human–carnivore conflict (Woodroffe 2000, Ripple et al. 2014). | Distance Accumulation tool producing a continuous 'cost' raster | 2.45 km (0.21 – 10.85 km) | 0.93 km (0.22 – 2.73 km) |
| Extent of recently burnt (%) | Dingoes have been found to positively respond to recently burnt areas in this region (Geary et al. 2018), and elsewhere (Bliege Bird et al. 2018). Relationships have been linked to habitat openness and numbers of preferred prey items (Catling et al. 2001). | Area of the recently burnt vegetation (0 -11years post-fire). | 12.86 (0 – 100) | 18.53 (0 - 100) |
| Extent of long unburnt (%) | Much of the oldest mallee vegetation in the region exists in isolated, long-unburnt fragments—a structural attribute identified as key for threatened vertebrate persistence (Takach et al. 2022). Currently there is little information linking dingoes explicitly to long-unburnt habitat. | Area of the late fire age class (> 35 years post-fire) | 35.62 (0 -100) | 35.96 (0 – 100) |
| Fire diversity (0 – 1) | Previous work in the region has indicated that a desirable mix of fire histories for biodiversity (birds, small mammals and reptile) conservation includes a combination of early, mid and late post-fire age-classes, weighted toward late seral stages (Kelly et al. 2012, Taylor et al. 2013, Giljohann et al. 2017). | Shannon diversity of fire age classes, based on the extent of recently burnt (0–11 years post-fire), mid (11 - 35 years post-fire), and late (>34 years post-fire) | 0.43 (0-1) | 0.67 (0 -1) |

| | | | | |
|---|---|---|---|---|
| | The influence of pyrodiversity on dingo distribution is less studied. | | | |

## Fire variables

Fire history and vegetation digital overlays were determined through 65 years (1958–2023) of LANDSAT satellite imagery state agency records (Department of Energy, Environment and Climate Action 2024). Three fire age classes were used to represent key post-fire successional stages in semiarid, Mallee ecosystems (Haslem et al. 2011), including recently burnt (< 11 years post fire), Mid-successional vegetation (11–35 years post-fire) and long-unburned vegetation (>35 years post-fire). These age classes were selected as they represent key post-fire successional stages in semiarid, Mallee ecosystems (Haslem et al. 2011, Department of Environment, Land, Water & Planning, Victoria 2025).

To represent the areas fire history, we used three fire age class variables, the extent of recently burnt and long unburnt vegetation within a 2,5 km moving window along with the diversity of fire age classes. Shannon's Diversity Index (SHDI) was calculated using:

$$SHDI = -\sum_{i=1}^{m} (P_i \ * lnP_i \ )$$

Where $P_i$ is the extent of class $i$ and $m$ is the number of classes, and $ln$ the natural logarithm. The extent of mid-successional vegetation was not included as it is negatively and highly correlated with the extent of recent and long unburned vegetation.

## Data Analysis

All predictor variables were continuous and, prior to statistical analysis, were standardised by subtracting the mean and dividing by two standard deviations (following Gelman 2008) to allow comparison of regression coefficients and interpretation of effects. We assessed spatial autocorrelation by constructing spline correlograms of the Pearson residuals (Bjørnstad and Falck 2001) using 1,0000 permutations for all individuals (marked and unmarked) and marked individuals only using the R-package ncf (Bjørnstad 2016).There was no evidence of spatial autocorrelation (Brooks and Gelman 1998) (Figure S1). Correlation between covariates; distance to roads, distance to water, extent of recently burnt and long unburnt vegetation, and diversity of fire age classes were weak (Pearson correlation coefficient <0.6), and were therefore all retained in the final model.

We used a Spatial Mark Recapture (SMR) model that used data from both marked and unmarked individuals to estimate dingo density and abundance. We fitted a discrete-space version of the SMR model used by (Forsyth et al. 2019, Bengsen et al. 2022), which uses data augmentation of the number of ‘marked’ and ‘unmarked’ individuals within a Bayesian framework (Chandler and Royle 2013). We augmented the number of potential marked and unmarked individuals to 100 as visual inspection of posterior distributions confirmed these values were sufficient to avoid truncation of abundance estimates (Chandler and Royle 2013, Royle et al. 2014). The state space was defined as a boundary of the park roughly encompassing the locations of cameras detecting dingoes (Fig 1) giving an area of 5,515 km$^2$. This state space was then discretised using a resolution of 2.5km with the locations of each camera assigned to a cell. To account for the effects of landscape variables on abundance and density we calculated values for distance to water, distance to roads, extent of long unburnt and fire diversity for each 2.5 km cell and included these values for all camera locations in our model. These covariates potentially influence the spatial distribution of individual dingo activity centres thereby altering expected number of individuals per cell. Model parameters were estimated using Markov chain Monte Carlo (MCMC) sampling in NIMBLE version 0.6–12 (NIMBLE Development Team 2017). To fit the model, we drew 60,000 samples from each of 3 MCMC chains, using diffuse initial values. We applied a thinning rate of 5 and discarded the first 20,000 samples as burn-in, leaving 8,000 samples from each chain to form the posterior distribution of the parameters, resulting in a total of 24,000 posterior samples. Convergence of the chains was assessed using the Brooks–Gelman–Rubin convergence statistic ($\hat{R}$) (Brooks and Gelman 1998) as well as visual inspection of traceplots (Figure S2).

Finally, the model parameters were used to predict spatial variation in dingo density across the state-space. This was undertaken by predicting dingo density for each pixel in the state-space using the environmental predictors for each pixel and the posterior distribution of the corresponding parameters. A total of 1000 posterior samples of the parameters were used to construct the spatial distribution of dingo density.

# Results

## Dingo data

We detected a total of 20 distinctly marked dingo individuals at a total of 30 unique locations, and unmarked dingoes at 34 unique camera locations, over 7,923 camera-days (Table S1). We identified eight individual dingo pups. Pups can have different activity centres and movement patterns than adults (Thomson et al. 1992) and were therefore removed from the analysis. Marked adult dingoes were detected a total of 249 times, with a median detection rate of 10.5 (range 1 – 39). Movements travelled by individuals varied from 0.00 km over a day to 119.83 km  km over 300 days (Table S2).

## Estimating dingo population size and density

The estimated abundance ($\hat{N}$) of dingoes within the state space (5,515 km$^2$) Abundance was estimated at 77 dingoes (95% CrI = 64–94; Fig. 2A), corresponding to a density of 0.014 dingoes/km$^2$ (95% CrI = 0.012–0.017; Table 2). The estimated spatial scale parameter ($\sigma$) was 7.64 km (95% CrI = 6.95–8.40; Table 2), equivalent to an average 95% area of activity (s) of 924 km$^2$ (Royle et al., 2013; Forsyth et al., 2019). The daily encounter rate ($\lambda$) when a camera coincided with the centroid of a dingo activity centre was 0.0060 (95% CrI = 0.0048–0.0075; Table 2).

Gelman–Rubin statistics were 1 for all parameters, and visual inspections of the trace plots indicated that the Markov chains successfully converged (Figure S2). Effective sample sizes exceeded 1,000 for all parameters (4,609–18,645), indicating good precision in the parameter estimates (Table 1).

## How does density vary in relation to landscape features

Dingo density was spatially variable across the region and decreased with increasing distance from water sources (mean = –3.40, 95% CrI = –5.51 – –1.69). Density was also positively related to distance to roads (mean = 2.20, 95% CrI = 1.35 – 3.03) (Fig. 4). Additionally, density was positively associated with the extent of long-unburnt areas (mean = 2.03, 95% CrI = 0.82 – 3.45) and with fire diversity (mean = 2.72, 95% CrI = 1.34 – 4.11), while it was negatively associated with the extent of recently burnt areas (mean = –3.23, 95% CrI = –5.94 – –0.72) (Fig. 4). Predicted dingo density based on landscape covariates indicated highest values in the central and eastern portions of the study area, with very low densities along the western margin (Fig. 3).

Table 2. Summary of posterior distributions of dingo abundances $\widehat{N}$ and densities $\widehat{D}$ (number of dingoes/km2) estimated from the SMR model. We also provide information for the detection function, detection probability ($\lambda_0$) and spatial scale (σ).

| Parameter | Estimate | Median (50%) | SD | 2.5% Credible limit | 97.5 % Credible limit | Effective sample size |
|---|---|---|---|---|---|---|
| $\widehat{N}$ | 77.10 | 76.00 | 7.89 | 64.0 | 94.0 | 12,207 |
| $\widehat{D}$ | 0.014 | 0.0138 | 0.0014 | 0.0116 | 0.0170 | 12,207 |
| σ | 7.64 | 7.63 | 0.37 | 6.95 | 8.40 | 18,645 |
| $\lambda_0$ | 0.0060 | 0.0060 | 0.0007 | 0.0048 | 0.0075 | 11,185 |

Figure 3. Spatial probability distribution of relative dingo occurrence across the Big Desert Wilderness Area, Victoria, Australia, over an approximate 12-month period (8 February 2023 – 15 January 2024). The distribution was modelled from posterior predictions of dingo density based on landscape covariates.

# Discussion

Robust population estimates for large carnivores and detailed knowledge of what environmental variables and landscape features influence variation in population abundance and density are poorly quantified for most species. Such information is vital, however, as understanding the ecological roles of large carnivores and how environmental factors influence population density can help guide more effective management and conservation actions.  We employed strategically placed camera traps in combination with a flexible spatial mark–resight modelling framework to overcome challenges associated with detecting and estimating populations of elusive carnivores (dingoes) in a remote semi-arid ecosystem. By linking spatially explicit population estimates to landscape variables—including, water availability, habitat structure and aspects of the fire regime —we were able to better understand how dingo density varied across the landscape. This information will help to guide refinement of dingo monitoring and management. Importantly, our approach can be adapted for surveying low density populations of large carnivores elsewhere.

Recent work has shown that dingoes in the Big Desert-Wyperfeld region exhibit significant inbreeding and that they are the most genetically isolated group of dingoes in Australia, on a trajectory towards local extinction (Weeks et al., 2024). Our population density estimate of 0.014 per km 2 (77 individuals), is similar to some recent estimates (0.06, mesic southeastern Australia, Forsyth et al., 2019) but much lower than several other previous estimates for dingoes (Mcilroy et al. 1986, Thomson et al. 1992, Corbett 1995, Allen 2012b). In addition the activity areas calculated in this study (924 km2) are much larger than previously reported by either Forsyth et al. (2019;69 km$^2$) and Gabriele-Rivet et al. (2020;30 km$^2$). This may be a be both a consequence of the need to access key resources such as water (see table S2. for distances moved for identifiable individuals) and/or the possible effects of lethal control on dingo demography, movements and territories (Allen 2015).

Our results indicate non-random landscape use by dingoes, as has been found for other large carnivores (Dickson et al. 2005, Pereira et al. 2012), with significant relationships detected for all environmental covariates. This has important implications not only for monitoring and conservation efforts but also for understanding spatial patterns of prey and mesopredator distribution and habitat use (Wooster et al. 2022). In line with previous findings in semi-arid and arid environments, our results indicate that dingo density varied significantly and negatively with an increasing distance to water. Dingoes are considered water-limited predators (Allen 2012) and in semi-arid regions are therefore more likely to be present closer to available water sources (Newsome et al. 2013).

Dingoes also showed a significant and consistent positive response of to an increased distance to roads. Large carnivores, including jaguars, pumas and wolves, have been found to preferentially use roads to varying extents (Harmsen et al. 2009, Whittington et al. 2011, St-Pierre et al. 2022, Dickie et al. 2022), while others such as lynx and bears (black, brown and grizzly) have been found to avoid roads (Brody and Pelton, 1989; Dixon, 1997; Suring et al., 1998; Basille et al., 2013; Grilo et al., 2015). In other parts of Australia dingoes are known to use roads, creeks and fence lines to more easily traverse landscapes Raiter et al. 2018). Vehicle tracks can enhance predator movement efficiency, potentially increasing hunting success and altering predator-prey dynamics. The low density of dingoes in our study region may mean that individuals have less need to exploit linear infrastructure or they are being actively avoided due to a perceived risk e.g. vehicle presence or lethal control (Mason et al. 2024), such as that observed in wolves (Latham et al., 2011 Mech et al., 1988).

Apex predators are likely to select habitats based on the location and abundance of preferred prey and in sections of the landscape where those prey are easiest to capture (Hopcraft et al. 2005, Milakovic et al. 2011, Cristescu et al. 2019, Wysong et al. 2020a). Much of the oldest mallee vegetation in the region is in isolated, long-unburnt fragments, representing relatively open understory, which is easier to navigate than the thick understorey of mid fire age class vegetation and the former also provides more shaded microclimatic refugia (aiding thermoregulation in a region where summer temperatures regularly exceed 40°C) compared to recently burnt areas. In Mallee ecosystems recently burnt vegetation has a simpler vegetation structure that supports major prey species of the dingo, including eastern (*Macropus giganteus*) and western grey kangaroos (*M. fuliginosis*) and swamp wallabies (*Wallabia bicolor*) (Davis et al. 2015, Mason et al. 2025a). Dingoes have been found to positively respond to recently burnt areas in this region, potentially suppressing the activity of foxes (Geary et al. 2018), and have demonstrated positive responses to recent burns elsewhere (Bliege Bird et al. 2018). In addition, in other systems dingoes have demonstrated decreasing abundance with increasing time since fire, likely associated with declines in large (predominantly macropod) prey abundance and habitat openness with increasing time since fire (Catling et al. 2001).

The negative response of dingo density to recently burnt vegetation and positive response to long unburnt vegetation in our study, is interesting, and possibly related to the scale at which individuals interact with their environment. As highly mobile, generalist predators, dingoes may select for areas that offer a range of fire age classes (see (Nimmo et al. 2018). Our results, along with recent dietary studies (Mason et al. 2025a), support the finding that dingoes likely require a range of fire age classes in the landscape. The negative association between dingo density and recently burnt vegetation may not reflect a direct avoidance of recently burnt habitat but rather a response to the broader landscape context. For example, recently burnt areas in this region are more commonly located near roads (Pearson's r = -0.44) due to fuel reduction burning, and further from permanent water sources (Pearson's r = -0.42).

The influence of the spatial and temporal patterns of fire is also influenced by surrounding land context, environmental gradients and particularly in arid and semi – arid regions recent rainfall history (Avitabile et al. 2013, Clarke et al. 2021). For example, rainfall patterns within Big Desert Wilderness area can be strongly influenced by El Nino Southern Oscillation (ENSO) phases (Noble and Vines 1993). During the survey period, the ENSO phase transitioned from a La Niña event, which concluded in early 2023, to neutral conditions. Increased water availability in the landscape may have influenced how social predators like dingoes utilised the area, with patterns of space use driven by both resource availability and socio-spatial interactions.

Dingoes have also been observed to exploit their environment to aid social learning of pups using vegetation or rocky outcrops to slow down prey (Pollock et al. 2022). Similarly, operating in the vicinity of a waterholes to target drinking kangaroos (Shepherd 1981, Purcell 2010). Additionally, temporal partitioning of waterhole access among individual dingoes has been documented in the Strzelecki Desert, suggesting structured resource use likely influenced by social hierarchy (Allen 2012b). Socio-spatial patterns of resource use are also observed in other social predators. For example, including African lions that concentrate around water and landscape features to increase prey vulnerability (Valeix et al. 2010) arctic foxes whose habitat selection reflects territoriality and parental care (Grenier-Potvin et al.

2021), and gray wolves that vary pack cohesion seasonally in their space use (Benson and Patterson 2015).

# Management implications

Our research successfully estimated the abundance and density of a vulnerable dingo population across a large, remote and difficult to access landscape. Importantly, our statistical approach was successful in estimating, with high certainty, the density and abundance of a wide ranging, hard-to-detect carnivore. Our approach could be modified and applied for other low density carnivore populations in remote and difficult to access landscapes elsewhere.

Fire-driven shifts in habitat structure also influence predator-prey dynamics, for herbivores and mesopredators, including some invasive species. Herbivores, including kangaroos (Meers and Adams 2003), are often attracted to recently burnt areas to graze on vegetation regrowth whereas mesopredators, including foxes, may avoid these areas due to the presence and potential fear of larger predators (Geary et al. 2018). Furthermore, fire-induced habitat openness may enhance predator hunting success, leading to localized increases in invasive mesopredator activity. Our findings emphasize the need for fire and wildlife management to take a whole-of-ecosystem approach that considers how species interactions between predators and prey might be shaped by variation in habitats and the availability and configuration of resources within landscapes.

The influence of disturbances affecting predator dynamics have significant implications for management. Our results indicate that human disturbances may influence dingoes both locally in the maintenance of vehicle tracks and prescribed burns, and globally due to the impacts of climate change on weather and water availability (Dore, 2005). With drought periods becoming more prolonged and severe, and changes in fire regimes (more frequent and severe fires) (Abatzoglou et al., 2019; Kelly et al. 2020; O'Donnell et al., 2014), the consideration of the spatial and temporal availability of water, and the extent and diversity, of fire-generated habitat variation, will be important for guiding dingo management and conservation and reducing conflict at the public-private land interface. Maintaining suitable habitat, with enough water and natural prey sources while and minimising future disturbances and though time poses on-going challenges for managing this vulnerable population. The extremely low density of individuals and the distances moved may also limit the success of genetic rescue or attempts to arrest the decline of genetic diversity, due to low encounter rates, or the strong social nature of the resident population. A better understanding of social interactions at important resources such as carcasses or water would likely aid monitoring and assessing any future management interventions.

Key recommendations include:

Ensuring access to water with spatial and temporal availability that supports a preferred prey population.

Protecting long - unburnt vegetation, in conjunction with a diversity of age classes especially near water sources.

Minimisation of new track creation to prevent further fragmentation of dingo habitat.

# Figures

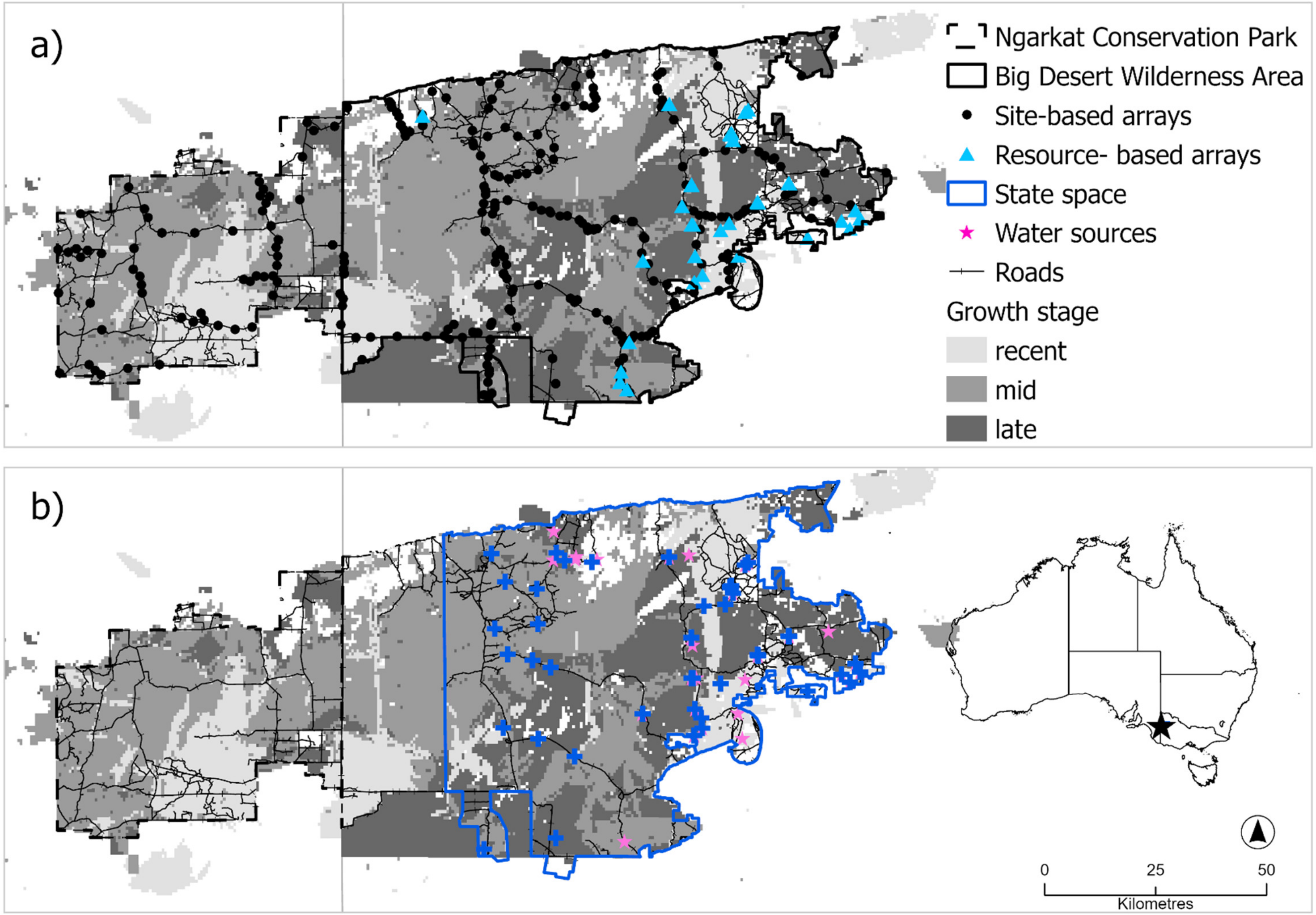
a)
Ngarkat Conservation Park
Big Desert Wilderness Area
Site-based arrays
Resource- based arrays
State space
Water sources
Roads
Growth stage
recent
mid
late
b)
0
25
50
Kilometres

Figure 1. Location of site and resource-based camera arrays traps in Big Desert Wilderness Area. Panel a) indicates all sampled sites. Panel b) indicates sites where dingoes were detected within the 12-month survey period.

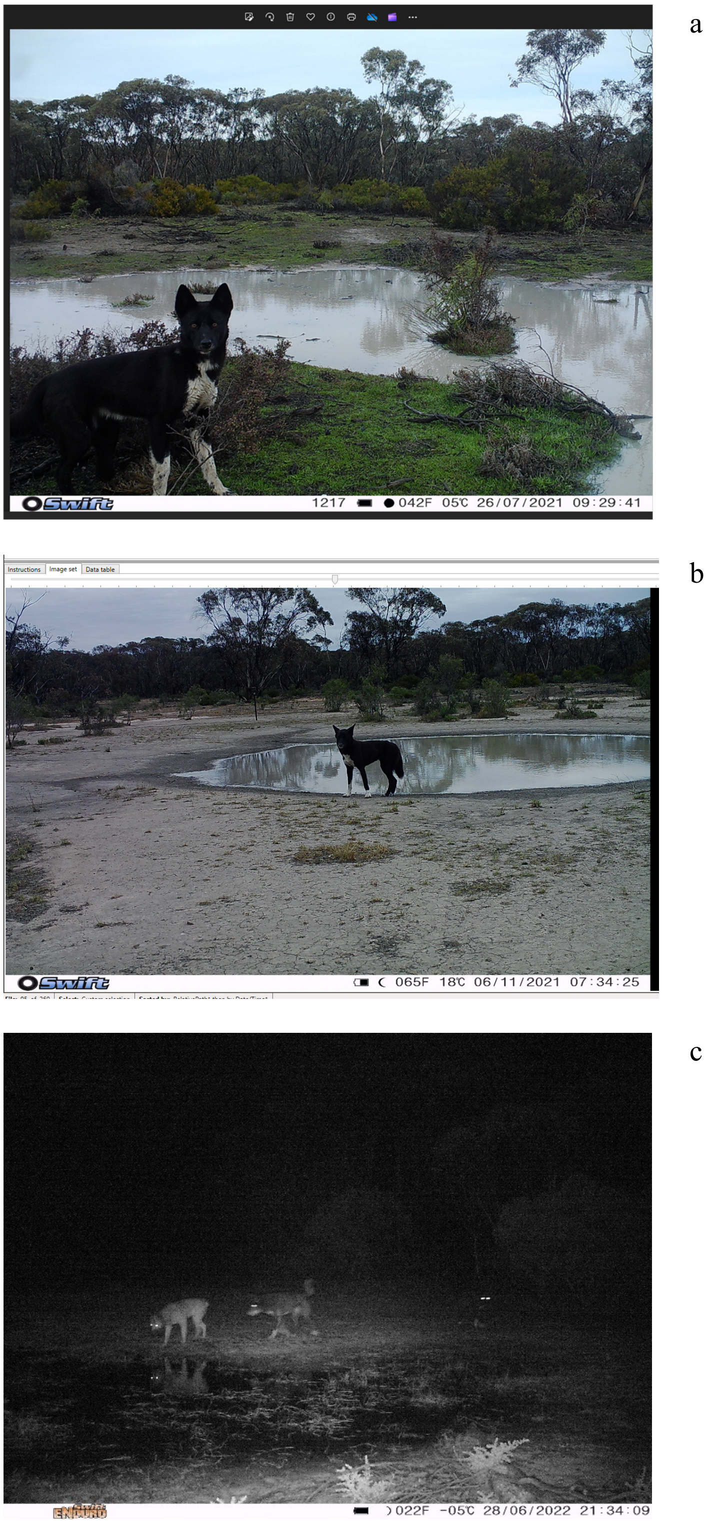

Figure 2. Three typical camera-trap images of dingoes (Wilkerr) in northwestern Victoria, Australia. Image a and image b indicate similar marked individuals which can be separated by length and shape of front leg 'socks'. White chest pattern on the top individual is also extended while the button individual stops more abruptly. Image c contains three individuals with some patterning visible, however markings are not sufficient to reliably separate them.

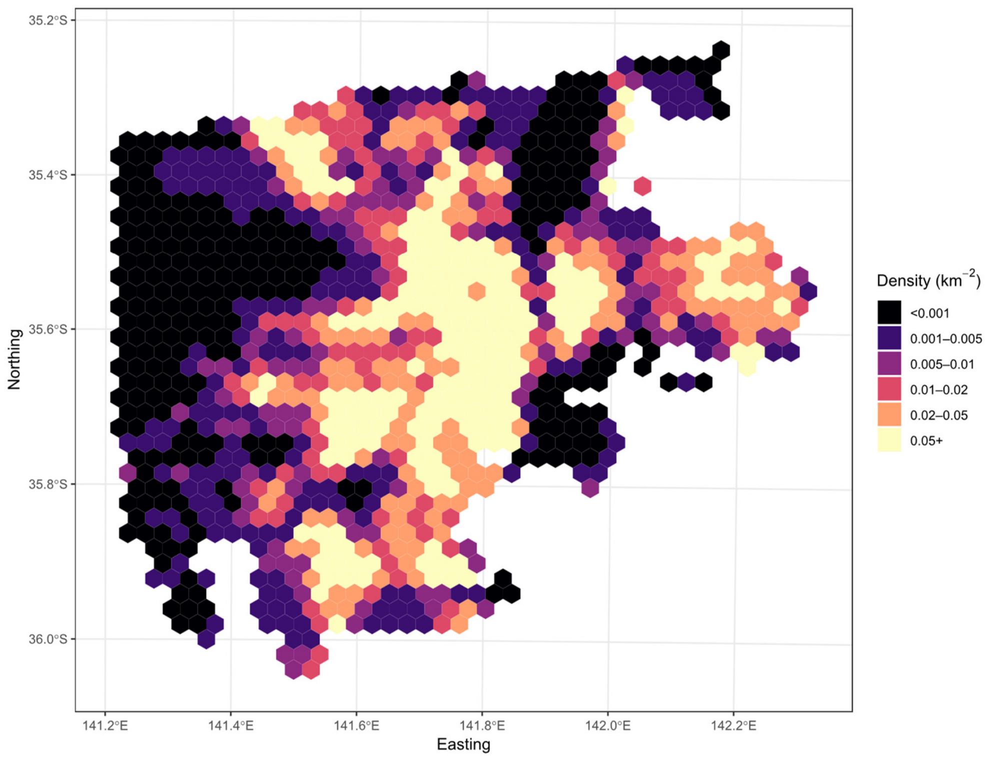

Figure 3. Spatial probability distribution of relative dingo occurrence across the Big Desert Wilderness Area, Victoria, Australia, over an approximate 12-month period (8 February 2023 – 15 January 2024). The distribution was modelled from posterior predictions of dingo density based on landscape covariates.

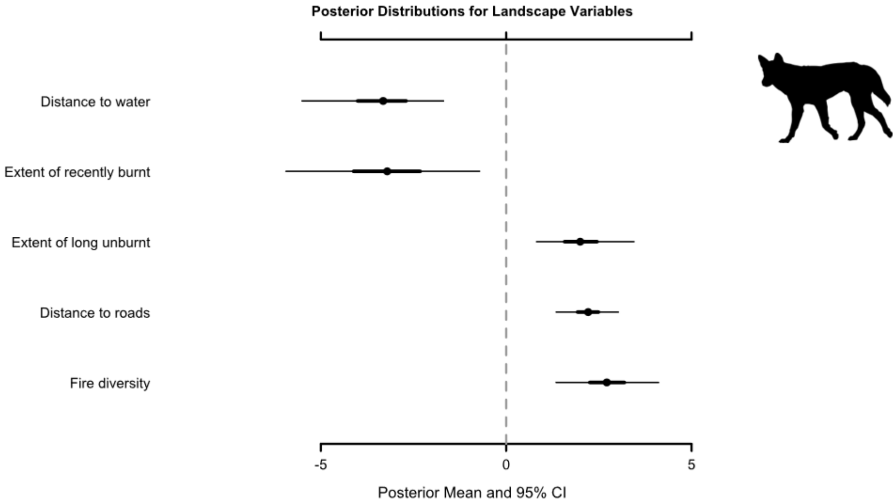

Figure 4. Posterior distributions for environmental covariates showing the influence of landscape variables on dingo density and abundance. The central dot represents the posterior mean, the thick line shows the 60% credible interval, and the thin line shows the 95% credible interval.

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# Supplementary

## Tables

### Table S1 - Final data set used in the SMR model

| Site type | Site ID | Start date | End date | No. of days a dingo was seen | Survey days |
|---|---|---|---|---|---|
| resource based | AG-site-4-cam1 | 1/01/2023 | 2/02/2023 | 1 | 33 |
| resource based | AG-site-5 | 1/01/2023 | 10/02/2023 | 11 | 44 |
| resource based | site-10-C006 | 11/06/2023 | 4/10/2023 | 3 | 116 |
| resource based | site-100-C082 | 10/12/2023 | 21/01/2024 | 20 | 54 |
| resource based | site-101-C005 | 17/10/2023 | 27/12/2024 | 1 | 438 |
| resource based | site-102-C052 | 11/12/2023 | 10/02/2024 | 0 | 62 |
| resource based | site-103-C006 | 6/10/2023 | 8/12/2023 | 0 | 64 |
| resource based | site-104-C009 | 23/10/2023 | 8/02/2024 | 0 | 109 |
| resource based | site-11-C010 | 19/06/2023 | 4/10/2023 | 0 | 108 |
| resource based | site-12-C027 | 21/06/2023 | 16/08/2023 | 0 | 57 |
| resource based | site-125-S011 | 1/01/2023 | 2/03/2023 | 0 | 61 |
| resource based | site-13-C028 | 22/05/2023 | 5/10/2023 | 2 | 137 |
| resource based | site-131-S021 | 1/01/2023 | 6/03/2023 | 0 | 65 |
| resource based | site-132-S022 | 1/01/2023 | 5/03/2023 | 0 | 64 |
| resource based | site-137-S027 | 1/01/2023 | 6/03/2023 | 0 | 65 |
| resource based | site-14-C033 | 5/06/2023 | 21/10/2023 | 0 | 139 |
| resource based | site-140-S030 | 1/01/2023 | 6/03/2023 | 0 | 65 |
| resource based | site-145-S035 | 1/01/2023 | 19/01/2023 | 0 | 19 |
| resource based | site-147-S037 | 1/01/2023 | 5/03/2023 | 0 | 64 |
| resource based | site-148-S038 | 1/01/2023 | 6/03/2023 | 0 | 65 |
| resource based | site-149-S039 | 1/01/2023 | 13/01/2023 | 0 | 13 |
| resource based | site-15-C029 | 11/06/2023 | 15/09/2023 | 3 | 98 |

| | | | | | |
|---|---|---|---|---|---|
| resource based | site-150-S040 | 1/01/2023 | 1/02/2023 | 0 | 32 |
| resource based | site-152-S042 | 1/01/2023 | 5/03/2023 | 0 | 64 |
| resource based | site-153-S043 | 1/01/2023 | 4/03/2023 | 0 | 63 |
| resource based | site-158-S048 | 1/01/2023 | 6/03/2023 | 0 | 65 |
| resource based | site-16-C029 | 12/06/2023 | 23/09/2023 | 0 | 104 |
| resource based | site-161-S051 | 1/01/2023 | 10/02/2023 | 0 | 41 |
| resource based | site-17-C047 | 26/06/2023 | 2/09/2023 | 0 | 69 |
| resource based | site-18-C026 | 2/07/2023 | 2/09/2023 | 1 | 63 |
| resource based | site-184-S074 | 1/01/2023 | 8/03/2023 | 0 | 67 |
| resource based | site-189-S079 | 1/01/2023 | 10/02/2023 | 0 | 41 |
| resource based | site-19-C041 | 9/07/2023 | 22/10/2023 | 6 | 107 |
| resource based | site-20-C093 | 16/07/2023 | 22/10/2023 | 19 | 106 |
| resource based | site-21-C095 | 10/07/2023 | 15/10/2023 | 2 | 98 |
| resource based | site-22-C097 | 10/07/2023 | 15/10/2023 | 3 | 98 |
| resource based | site-24-C076 | 23/07/2023 | 27/08/2023 | 0 | 36 |
| resource based | site-26-C063 | 30/07/2023 | 21/10/2023 | 0 | 84 |
| resource based | site-27-C086 | 30/07/2023 | 21/10/2023 | 0 | 84 |
| resource based | site-28-C054 | 30/07/2023 | 21/10/2023 | 0 | 84 |
| resource based | site-29-C051 | 31/07/2023 | 15/10/2023 | 5 | 78 |
| resource based | site-30-C091 | 14/08/2023 | 23/09/2023 | 5 | 43 |
| resource based | site-31-C068 | 14/08/2023 | 23/09/2023 | 10 | 45 |
| resource based | site-32-C092 | 14/08/2023 | 23/09/2023 | 4 | 41 |
| resource based | site-33-C073 | 4/09/2023 | 17/09/2023 | 0 | 14 |
| resource based | site-34-C096 | 4/09/2023 | 17/09/2023 | 0 | 14 |
| resource based | site-35-C067 | 10/09/2023 | 4/10/2023 | 4 | 26 |
| resource based | site-36-C040 | 10/09/2023 | 4/10/2023 | 3 | 26 |
| resource based | site-37-C036 | 17/10/2023 | 28/10/2023 | 0 | 12 |
| resource based | site-39-C001 | 19/06/2023 | 29/07/2023 | 0 | 41 |

| | | | | | |
|---|---|---|---|---|---|
| resource based | site-40-C004 | 22/05/2023 | 23/07/2023 | 0 | 63 |
| resource based | site-41-C008 | 19/06/2023 | 29/07/2023 | 0 | 41 |
| resource based | site-42-C009 | 16/07/2023 | 22/10/2023 | 11 | 100 |
| resource based | site-43-C038 | 26/06/2023 | 2/09/2023 | 0 | 69 |
| resource based | site-44-C-098 | 16/07/2023 | 22/10/2023 | 8 | 100 |
| resource based | site-45-C-094 | 16/07/2023 | 22/10/2023 | 11 | 102 |
| resource based | site-46-C-088 | 16/07/2023 | 22/10/2023 | 4 | 99 |
| resource based | site-51-C-066 | 23/07/2023 | 27/08/2023 | 10 | 39 |
| site based | site-53-S050-C087 | 1/01/2023 | 5/03/2023 | 4 | 65 |
| site based | site-54-S053-C017 | 1/01/2023 | 6/03/2023 | 0 | 65 |
| site based | site-58-S083-C027 | 1/01/2023 | 8/03/2023 | 2 | 68 |
| resource based | site-65-C001 | 17/12/2023 | 21/01/2024 | 6 | 39 |
| resource based | site-66-C002 | 29/01/2024 | 10/02/2024 | 1 | 13 |
| resource based | site-67-C007 | 5/10/2023 | 9/02/2024 | 0 | 128 |
| resource based | site-68-C008 | 16/10/2023 | 10/02/2024 | 0 | 118 |
| resource based | site-69-C028 | 7/10/2023 | 26/01/2024 | 17 | 116 |
| resource based | site-7-C003 | 11/06/2023 | 4/10/2023 | 2 | 116 |
| resource based | site-70-C019 | 5/10/2023 | 10/02/2024 | 9 | 131 |
| resource based | site-71-C047 | 3/09/2023 | 26/12/2023 | 0 | 115 |
| resource based | site-72-C038 | 3/09/2023 | 26/12/2023 | 0 | 115 |
| resource based | site-73-C026 | 3/09/2023 | 26/12/2023 | 0 | 115 |
| resource based | site-74-C041 | 11/10/2023 | 2/02/2024 | 15 | 118 |
| resource based | site-75-C093 | 23/10/2023 | 2/02/2024 | 18 | 107 |
| resource based | site-76-C095 | 16/10/2023 | 26/12/2023 | 0 | 72 |
| resource based | site-77-C097 | 16/10/2023 | 26/12/2023 | 0 | 72 |
| resource based | site-78-C098 | 23/10/2023 | 8/02/2024 | 69 | 157 |
| resource based | site-79-C024 | 3/02/2024 | 8/02/2024 | 32 | 32 |
| resource based | site-8-C004 | 24/07/2023 | 5/10/2023 | 0 | 74 |

| | | | | | |
|---|---|---|---|---|---|
| resource based | site-80-C094 | 23/10/2023 | 2/02/2024 | 88 | 152 |
| resource based | site-81-C088 | 23/10/2023 | 2/02/2024 | 45 | 124 |
| resource based | site-82-C066 | 28/08/2023 | 16/12/2023 | 31 | 119 |
| resource based | site-83-C076 | 28/08/2023 | 16/12/2023 | 0 | 111 |
| resource based | site-84-C059 | 11/12/2023 | 10/02/2024 | 0 | 62 |
| resource based | site-85-C003 | 5/10/2023 | 8/12/2023 | 0 | 65 |
| resource based | site-86-C051 | 16/10/2023 | 9/12/2023 | 0 | 55 |
| resource based | site-87-C091 | 24/09/2023 | 27/01/2024 | 0 | 126 |
| resource based | site-88-C068 | 24/09/2023 | 27/01/2024 | 0 | 126 |
| resource based | site-89-C092 | 24/09/2023 | 27/01/2024 | 0 | 126 |
| resource based | site-9-C005 | 17/10/2023 | 28/10/2023 | 0 | 12 |
| resource based | site-90-C039 | 28/08/2023 | 16/12/2023 | 0 | 111 |
| resource based | site-91-C067 | 5/10/2023 | 8/12/2023 | 0 | 65 |
| resource based | site-92-C004 | 6/10/2023 | 26/01/2024 | 0 | 113 |
| resource based | site-93-C040 | 5/10/2023 | 8/12/2023 | 4 | 66 |
| resource based | site-94-C036 | 17/10/2023 | 27/01/2024 | 0 | 103 |
| resource based | site-95-C081 | 29/10/2023 | 27/01/2024 | 0 | 91 |
| resource based | site-96-C069 | 29/10/2023 | 27/01/2024 | 0 | 91 |
| resource based | site-97-C083 | 29/10/2023 | 27/01/2024 | 0 | 91 |
| resource based | site-98-C071 | 10/12/2023 | 21/01/2024 | 0 | 43 |
| resource based | site-99-C072 | 17/12/2023 | 21/01/2024 | 0 | 36 |

Table S2: distance (km) move by each identifiable individuals over time (days)

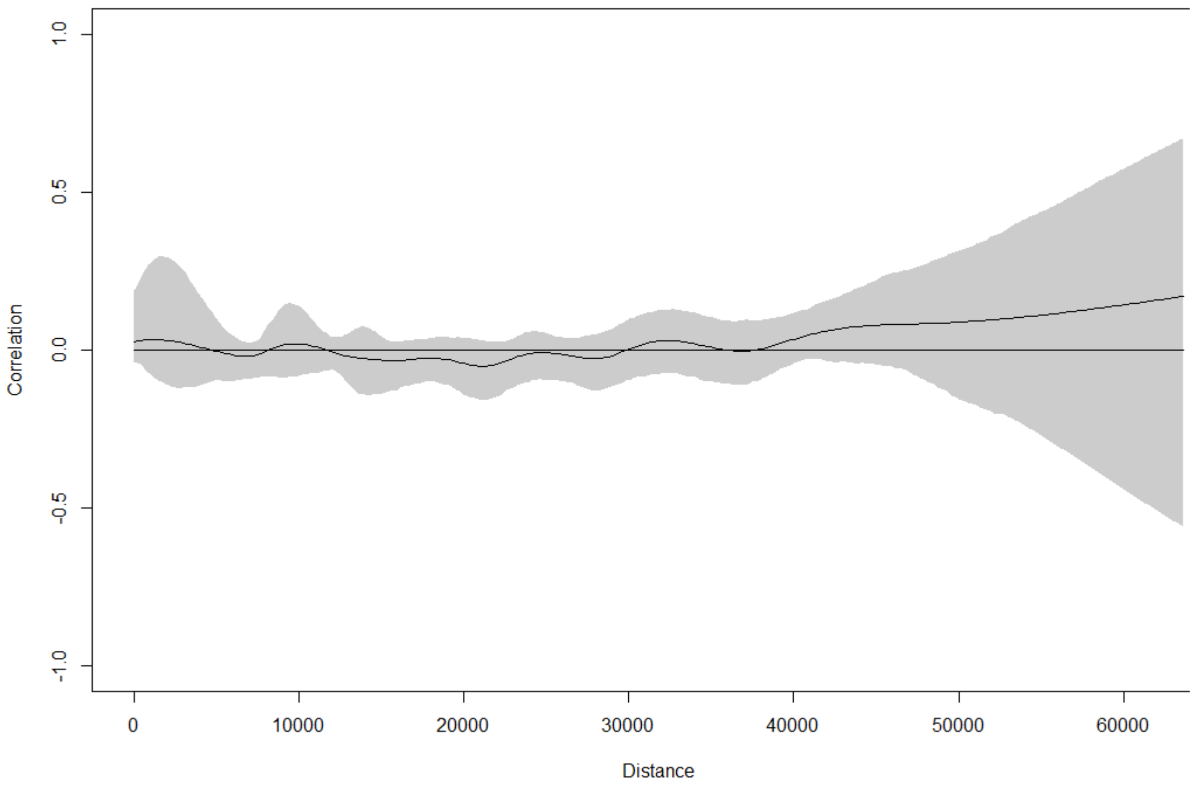
| ID | Total distance in Km | Total days | Number of observations |
|---|---|---|---|
| AGS1A1 | 0.00 | 0 | 2 |
| AGS2A1 | 56.15 | 185 | 77 |
| AGS3A1 | 119.83 | 300 | 68 |
| AGS4A1 | 86.53 | 305 | 114 |
| AGS5A1 | 92.73 | 295 | 70 |
| AGS5A2 | 88.52 | 259 | 21 |
| AGS5A3 | 48.38 | 302 | 83 |
| AGS5A4 | 49.03 | 295 | 76 |
| AGS6A1 | 0.02 | 34 | 8 |
| EMS100A1 | 82.94 | 142 | 20 |
| EMS13A1 | 96.19 | 87 | 10 |
| EMS19A1 | 0.00 | 0 | 1 |
| EMS20A1 | 82.67 | 122 | 22 |
| EMS20A2 | 115.94 | 120 | 30 |
| EMS42A1 | 0.00 | 53 | 2 |
| EMS44A1 | 86.16 | 70 | 14 |
| EMS65A1 | 0.00 | 0 | 2 |
| EMS7A1 | 75.12 | 123 | 18 |
| RMS53A1 | 0.00 | 15 | 4 |
| RMS82A1 | 0.00 | 88 | 8 |
| AGS1A1 | 0.00 | 0 | 2 |
| AGS2A1 | 56.15 | 185 | 77 |
| AGS3A1 | 119.83 | 300 | 68 |

Table S3: indices used to derive known water locations from Sentinel-2 Level 2A imagery seasonal stacks

| Sensor | Data extracted | Resolution | Formula | Band width (nm) | Reference |
|---|---|---|---|---|---|
| Sentinel-2 | B11 - Short-wave infrared (SWIR 1) | 20 m | | 1613.7 | |
| | B8 - Near-infrared (NIR) | 10 m | | 842 | |
| | B4 - Red | 10 m | | 665 | |
| | B3 – Green | 10 m | | 560 | |
| | B2 – Blue | 10 m | | 490 | |
| | Normalised Difference Water Index (NDWI) | | (G-NIR)/(G+NIR) | | (McFeeters, 1996) |
| | MNDWI | | (Green – SWIR2) / (Green + SWIR2) | | |
| | Modified Soil Adjusted Vegetation index (MSAVI-2) | | (2 × NIR + 1 – √((2 × NIR + 1)^2 – 8 × (NIR – RED))) / 2 | | (Qi et al., 1994) |

# Figures

(a)


1.0
0.5
0.0
-0.5
-1.0
Correlation
0
10000
20000
30000
40000
50000
60000
Distance

(b)

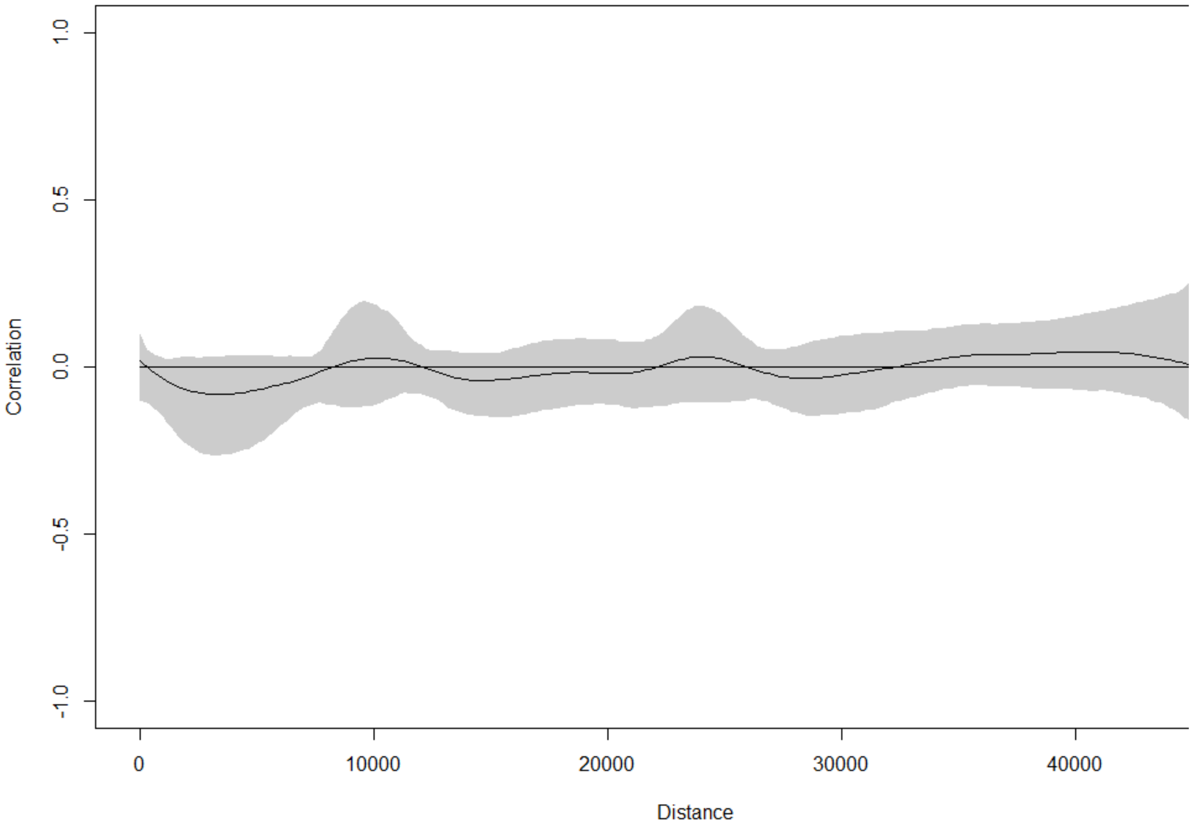
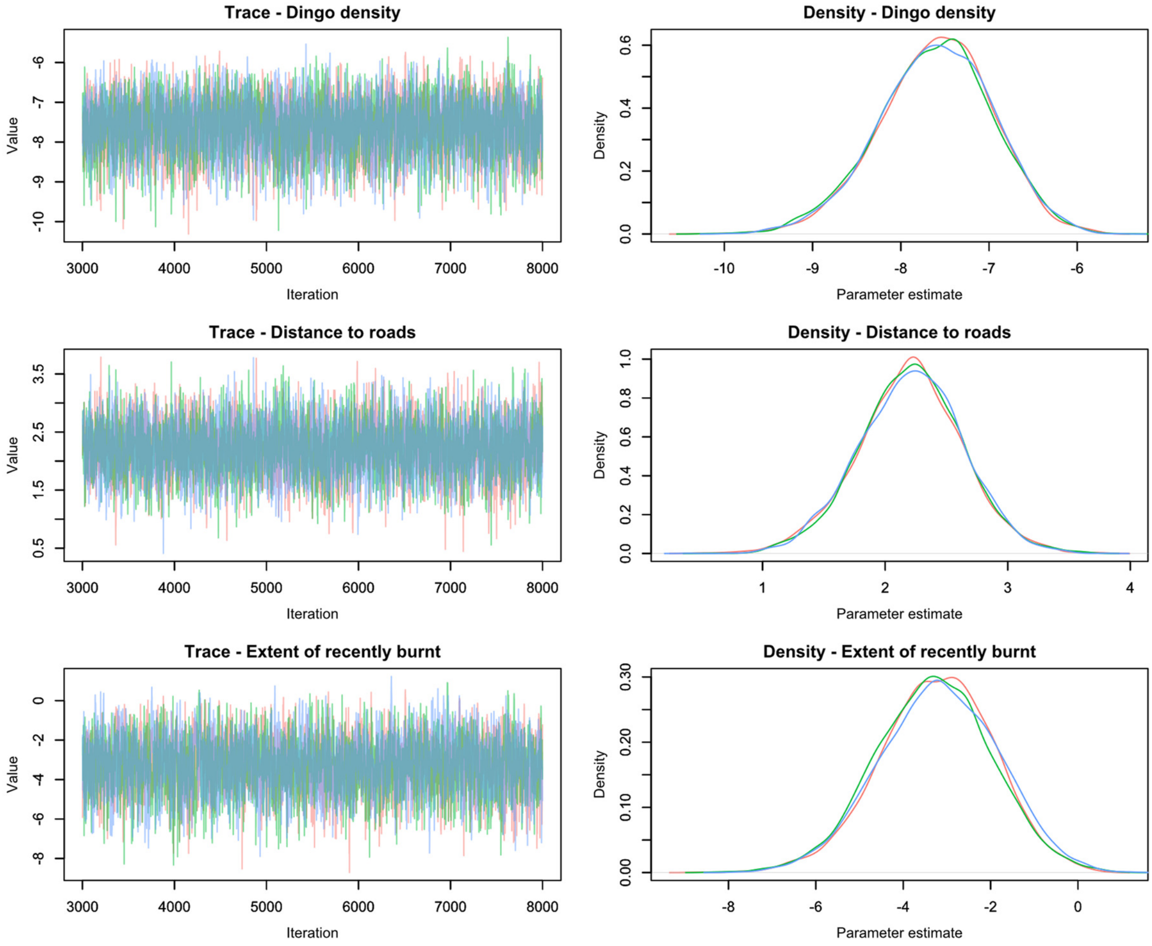


Figure S1: Spatial autocorrelation plots for the count of all individuals (a) and the count of marked individual only (b)


Trace - Dingo density
Value
-6
-7
-8
-9
-10
3000
4000
5000
6000
7000
8000
Iteration
Density - Dingo density
Density
0.6
0.4
0.2
0.0
-10
-9
-8
-7
-6
Parameter estimate
Trace - Distance to roads
Value
3.5
2.5
1.5
0.5
3000
4000
5000
6000
7000
8000
Iteration
Density - Distance to roads
Density
1.0
0.8
0.6
0.4
0.2
0.0
1
2
3
4
Parameter estimate
Trace - Extent of recently burnt
Value
0
-2
-4
-6
-8
3000
4000
5000
6000
7000
8000
Iteration
Density - Extent of recently burnt
Density
0.30
0.20
0.10
0.00
-8
-6
-4
-2
0
Parameter estimate

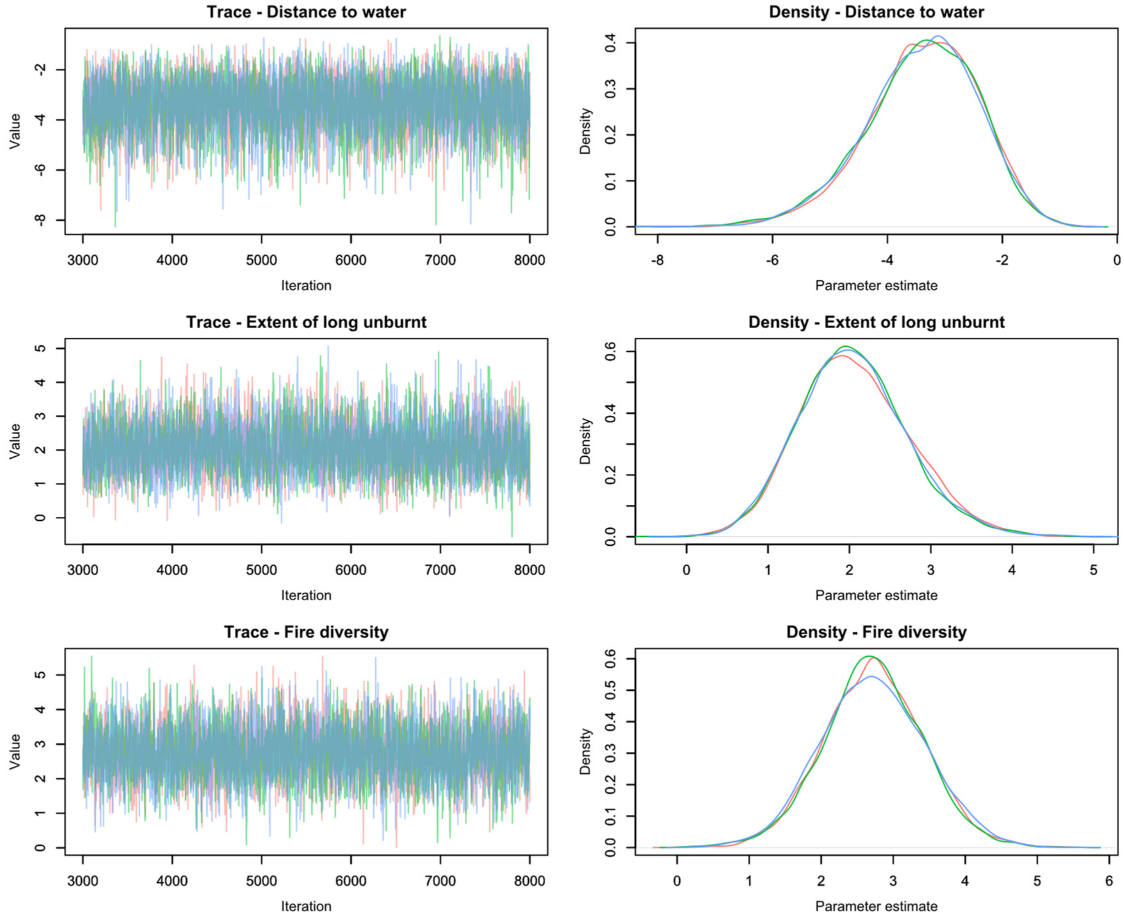


Figure S2 Trace plots for SMR Model